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Conservation challenges of the endangered gravel grasshopper *Chorthippus* (*Glyptobothrus*) *pullus* along Alpine rivers: Insights from remote sensing in Northwestern Italy

Luca Anselmo[1]

1 Department of Life Sciences and System Biology, University of Turin, 10123 Turin, TO, Italy.

Corresponding author: Luca Anselmo (luca.anselmo@unito.it)

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Abstract

Floodplains are considered among the most endangered land types on a global scale. Only a few natural and semi-natural floodplains remain in the European Alps and are currently considered threatened. This study focused on *Chorthippus* (*Glyptobothrus*) *pullus* (Philippi, 1830), a small grasshopper widely considered an indicator of well-preserved riverine habitats. In the European Alps, this species now survives alongside very few watercourses, showing an increasingly rarefied and fragmented distribution due to habitat loss, mainly caused by artificial flow regulation. Presence data were collected in the upper Susa Valley (NW Italy) along transects positioned in floodplains and constrained reaches of the Dora Riparia and Dora di Bardonecchia rivers. A species distribution model was implemented using topographic variables and Normalized Difference Vegetation Index derived from remote sensing. Suitability and connectivity maps were produced to identify key conservation areas. Furthermore, the times of colonization, persistence, and extinction of this species along the vegetation succession at the patch level were estimated. The study revealed the presence of a significant population of *C. pullus* in the study area, primarily distributed in large floodplains. Connectivity between suitable areas was limited by the regulation of certain river sections, particularly in urbanized areas. The results confirmed it as a pioneer species, which tends to become extinct in patches when an undisturbed succession reaches an unsuitable stage for the species in about 12 years from bare soil. According to the study results, the conservation of this species depends on the maintenance of large floodable surfaces, which create a mosaic of patches with varying flooding frequencies. Specific recommendations are provided for its long-term conservation in the upper Susa Valley.

Keywords

bioindicator, Caelifera, NDVI, maxent, riverbanks, riverine species, succession, Susa Valley

Introduction

Floodplains are relatively flat areas extending from the banks of a river over which water flows at times of high discharge, often dividing it into channels (Ward et al. 2002, Goudie 2004, Naimann et al. 2005). These are therefore surfaces formed by fluvial sediment aggradation (Brierley and Fryirs 2005, Krizek et al. 2006), hosting a complex mosaic of patches of habitats at different stages of succession (Ward et al. 1999a, Stanford et al. 2005, Tockner et al. 2010). In natural contexts, these dynamic systems comprise habitats dominated by species capable of surviving riverine activity, mainly characterized by a balanced alternation between floods and dry periods (Müller 1995, Ward et al. 1999a, 2000, Arscott et al. 2002, Gurnell et al. 2005, Naiman et al. 2005, Leuschner and Ellenberg 2017), and between the accumulation and erosion of sediments (Tockner et al. 2000, 2006, Prach and Walker 2020). Floods can completely remove previous communities, triggering primary succession on a new layer or triggering secondary succession either by flowing a layer down to a new location or by adding a new layer on top of existing soil (Prach and Walker 2020). Therefore, many floodplain organisms are pioneer species strictly dependent on flood processes, which periodically remove and create suitable habitats for their reproduction (Müller 1995, Scott et al. 1996, Müller and Scharm 2001, Camporeale et al. 2013). Natural floodplains can contain complex habitat mosaics that develop along a gradient of climate, soil moisture, disturbance, and nutrients (Ward et al. 1999a, 2002, Naiman and Décamps 1997, Stanford et al. 2005, Naiman et al. 2005), thus hosting high species diversity. Moreover, river environments are important elements that connect existing natural areas and improve the overall quality of ecosystems (Naiman et al. 1993, 2005, Ward et al. 1999a, Keruzore et al. 2013). However, floodplains are considered among the most endangered landform types worldwide (Tockner and Stanford 2002, Tockner at al. 2008). Human activities causing alteration of the hydrologic regime and functioning of the watercourses constitute a major threat to riverine habitats in Europe (Janssen et al. 2016), leading to the current conservation status in Europe being critical, as 95% of the original area of riverine habitat has been converted to anthropogenic uses (European Environment Agency 2018), and 70%–100% has been lost over the past century (European Environment Agency 2020). Only a few natural and semi-natural floodplains are left in the European Alps, but all of these are currently threatened (Müller 1995, Ward et al. 1999b, Gurnell et al. 2009, Comiti 2012). The main anthropogenic

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pressures are hydrological system modifications, construction of dams and barriers for energy production, water supply, and sand and gravel extraction (Janssen et al. 2016). When disturbed by human activities, riverine habitats are easily invaded by pioneer exotic plants (Janssen et al. 2016). Furthermore, climate change represents another potential threat because it can cause the permanent alteration of the water regime (Janssen et al. 2016). As a result of these pressures, species once dominant in floodplains are showing a rapid decline worldwide (Müller and Scharm 2001, Bilz et al. 2011).

This study focuses on *Chorthippus* (*Glyptobothrus*) *pullus* (Philippi, 1830), a small and rare grasshopper sparsely distributed in central Europe and across the Balkans up to the Caucasus (Harz 1975, Fontana et al. 2002, Massa et al. 2012). In Italy, this species is mainly linked to very sparse floodplains of mountain rivers and streams along the Alps, showing an extremely fragmented distribution (Fontana et al. 2002, 2004, 2005, Massa et al. 2012, Iorio et al. 2019, Tami et al. 2005, 2021). It lives in open and sunny habitats covered by gravel, silt, and sparse vegetation, such as heterogeneous riverine environments that include both more and less frequently flooded patches (Tami et al. 2021), where it finds the pioneer grass on which it mainly feeds (Schwarz-Waubke 1997, Tami et al. 2021). To date, this species has been observed in Italy between 160 m and 2000 m a.s.l. in the Eastern Alps and between 1000 m and 1700 m in the Western Alps (Tami et al. 2021). It can also be found in dry grasslands, heaths, and clearings far from water courses in Germany (Baur et al. 2006, Janssen 2003, Landeck and Nähring 2012). *C. pullus* is a low-mobility species due to its short wings (Kurth 2007, Maag et al. 2013, Hochkirch et al. 2016a). Their populations often have very low densities (Tami et al. 2021) and are typically found in small patches of less than 100 m^2 (Janssen 1993, Schwarz-Waubke 1998) and often of only 20–30 m^2 (Fontana 2004, Maag et al. 2013). In Italy, *C. pullus* exhibits summer-autumn phenology (Massa et al. 2012, Iorio et al. 2019), but the adults appear relatively early compared to other species (Tami et al. 2021). At 500–600 m of altitude in the Eastern Italian Alps, adults can be observed from the end of June until the beginning of September, and at lower altitudes, densities begin to reduce at the beginning of August (Tami et al. 2021). In Austria, half of the overall observations collected by Panrok (2017) were from August, but at least a quarter of the data came from before mid-July. In the two years of research conducted near Salzburg (Germany) by Schwarz-Waubke (2001), the highest density of adults was reached between mid-June and early July in 1993 and on July 1 in 1994. The presence of this species in Europe has been widely considered an indicator of well-preserved riparian ecosystems (Reich 1991, 1998, Fontana et al. 2002, 2004, Massa et al. 2012, Tami et al. 2005, 2021). This rare grasshopper is, in fact, only present in environments where the fluvial pulsations can act freely along the river channels, while it disappears where the regulation of water and the extraction of gravel have distorted the natural fluvial dynamics (Reich 1998, Hochkirch et al. 2016b, Landmann 2023). For this reason, and due to the anthropization of watercourses that has reduced its available habitats, *C. pullus* is considered threatened in many European countries (Detzel 1998, Ingrisch and Köhler 1998, Sardet and Defaut 2004, Baur et al. 2006, Monnerat et al. 2007, Hilpold et al. 2017, Landmann 2017, 2022), including Italy (Fontana et al. 2002, 2004, Tami et al. 2005, 2021). Loss of local habitat caused by flooding or localized and time-limited human acts (such as occasional gravel extraction and works aimed at protecting the banks) generally does not constitute a conservation concern for *C. pullus*, provided that new habitats with similar characteristics are available or can be quickly re-established in nearby areas and in continuity along watercourses (Tami et al. 2021).

The gravel grasshopper *C. pullus* was classified as Least Concern (LC) in the IUCN European Red List of Grasshoppers, Crickets and Bush-crickets (Hochkirch et al. 2016a), but is known to be declining in large parts of its range, primarily in western Europe (Hochkirch et al. 2016b). According to the IUCN assessment, the most important conservation action is protection of its habitats and restoration of natural river courses, mainly through maintenance of natural flood dynamics (Hochkirch et al. 2016b).

This study aimed to investigate the distribution and habitat requirements of *C. pullus* in the upper Susa Valley (NW Italy), one of the few areas of its presence in the western Italian Alps (Fontana et al. 2005, Sindaco et al. 2012, Tami et al. 2021), to provide recommendations useful for conservation planning. For this purpose, a species distribution model (SDM) that used data derived from remote sensing was employed. Satellite information has been gaining greater importance in ecological studies (Kerr and Ostrovsky 2003, Crowley and Cardille 2020, Wilson et al. 2020) to, for example, model the distribution of species (He et al. 2015, Randin et al. 2020, Schwager and Berg 2021) or to estimate vegetation changes over time (Meera Gandhi et al. 2015, Maynard et al. 2016, Zhao et al. 2020). SDMs are becoming increasingly essential not only for the identification of important conservation areas but also as a foundation for assessing landscape connectivity (Elith and Leathwick 2009). Connectivity is of crucial importance for species conservation, as it allows the movement of individuals between populations and promotes genetic diversity, propagule dispersal, and colonization of new habitats (Hanski and Ovaskainen 2000, Fahrig 2003). In assessing species conservation, the need to identify areas with poor connectivity is widely recognized (Beier and Noss 1998, Hilty et al. 2006, McRae et al. 2008). However, to ensure medium- and long-term conservation, the natural dynamics and management of the presence areas must be prioritized (Margules and Pressey 2000). Therefore, in this study, in-depth analyses were performed that focused on the relationship between the presence of this species and vegetation succession. In this framework, the concept of species persistence time (Bertuzzo et al. 2011, Suweis et al. 2012) was adapted to estimate the time between colonization and abandonment of a patch by the species along an undisturbed succession.

Material and methods

Study area.—The study was conducted in the upper Susa Valley (TO), located in the Cottian Alps of Northwestern Italy. This area falls within the Dora Riparia drainage basin, crossed by two main rivers: Dora Riparia and Dora di Bardonecchia. The basin covers a surface of 641.8 km^2, with an altitude range of ~307–3.131 m.a.s.l. Its land cover is mostly forest, with a prevalence of conifers. The upper Susa Valley is one of the driest areas of the western Alps, with mean annual rainfall of less than 600 mm/year at the center of the valley. The last extreme flood occurred in 2000, which extensively remodeled the upper Susa Valley riverine environments (Mercalli and Cat Berro 2018). The study area was restricted to the surface included in the "recent riverbed," identified by the GEmMA information system of river morphology provided by ARPA Piemonte (available at https://geoportale.arpa.piemonte.it), which compiles morphological and hydrological information on the main watercourses of Piedmont. It extends for ~2.52 km^2 and between 307 and 1915 m.a.s.l, intercepting some protected areas of the Natura 2000 network established in accordance with the Habitats Directive (92/43/EEC). The study area includes sectors where rivers are surrounded by variously sized natural or semi-natural floodplains and by constrained reaches with narrow riverbanks artificially shaped for flow regulation purposes in some cases.

Sampling design.—The field sampling of the species took place in July 2022. The month was selected based on the maximum density of adults expected: considering available information (summarized in the introduction section) and the elevation of the study area, I chose the first weeks of July to intercept a density of adults close to the maximum.

To obtain presence data rather than abundance data, and at the same time to investigate as many habitat patches as possible, I decided to use the transect sampling technique. Furthermore, I consider this method the most appropriate for quick and extensive investigation of riverine habitats after observing the limited effectiveness of other methods, such as sweep netting or rings, for low densities of short-winged grasshoppers in sparse vegetation (pers. obs.). The transects were of variable length (at least 250 m) and were placed parallel to the river and evenly spaced along the study area to cross patches of habitat that reflect as much as possible all the environmental conditions. They were placed at progressive distances from the river, at least 5 m apart from each other, in floodplains or in sufficiently wide constrained reaches. A stick was used to move the tufts of grass to spot the fleeing specimens disturbed by the rubbing of the grass and the approaching of the operator. It was not necessary to capture the specimens in most of the cases, as this short-winged species is not able to make long leaps, and its sighting and identification at the landing point was quite easy (pers. obs.). The speed traveled by the operator was low, at about 1.5 m/s. The specimens were identified in the field by their well-recognizable external morphology, based on Sardet et al. (2015) and Iorio et al. (2019). Each occurrence of *C. pullus* was georeferenced with a GPS system, as was the transect track, thus allowing knowledge of the spatial effort of the sampling across the study area. The hours between 11 a.m. and 15 p.m. of sunny days and with temperatures higher than 25 °C were chosen to maximize the likelihood of observing the species in activity (Maag et al. 2013). The most frequent plant species in the presence habitats were identified alongside the sampling of *C. pullus* to provide a generic description of the vegetational composition in which the species was present.

Environmental variables.—To model *C. pullus* distribution and environmental requirements, high-resolution environmental layers that I considered relevant for its ecology were utilized: Digital Elevation Model (DEM), Topographic Position Index (TPI), Terrain Ruggedness Index (TRI), and Normalized Difference Vegetation Index (NDVI). The DEM with an original resolution of 10 m (available at https://www.geoportale.piemonte.it) was used to derive the other two topographic variables TPI and TRI using the terra package (Hijmans 2023) in R v. 4.2.2 (R Core Team, 2020). The DEM was chosen because it can be considered the main proxy for temperature variation along elevation in mountain regions (Wilson and Gallant 2000). TPI describes the position of a pixel based on its elevation relative to the average of the surrounding cells (Jenness 2006). TRI represents the mean difference in elevation between a central pixel and its surrounding cells, expressed as the absolute value of the differences (Riley et al. 1999). These two indices were selected because they are potentially related to the amount and type of sediment accumulation. The NDVI is a radiometric measure of vegetation greenness, calculated using the red and near-infrared reflectance bands (Rouse et al. 1974). NDVI was chosen as is considered a proxy of vegetation productivity (Pettorelli et al. 2005, 2011): values between -1 and 0 correspond to areas ranging from water to bare soil, and the vegetation greenness increases up to the maximum value of 1 (Neigh et al. 2008, Wegmann et al. 2016). The NDVI layer was obtained using QGIS v. 3.16.14 (QGIS Development Team 2021), starting from red and near-infrared bands recorded by the Copernicus SENTINEL 2B satellite on 8 July 2022 (available at https://finder.creodias.eu), with a resolution of 10 m, without cloud cover on the study area (assessed by accurate visual inspection of the images) and concurrent with the sampling of the species. As an LC1 product type, atmospheric correction was applied using the Semi-Automatic Classification Plugin (SCP) in QGIS.

Species distribution modeling.—Given the small size and low density of *C. pullus* and the lack of sampling repetitions, absences would not be considered reliable. Therefore, a presence-based model was used for the analysis: the maximum entropy approach in MaxEnt (Phillips et al. 2006, Phillips and Dudík 2008) was chosen to model the species' distribution. This widely used algorithm involves the use of presence-only data and a set of environmental predictors (Phillips et al. 2006, Phillips and Dudík 2008, Baldwin 2009). MaxEnt compares the conditions in the presence locations with all available environments by sampling many points throughout the study area, which are referred to as background samples (Phillips et al. 2006, Phillips and Dudík 2008, Elith et al. 2011, Thibaud et al. 2014). This algorithm is often more efficient than other models (Elith et al. 2006, Guisan et al. 2007, Peterson et al. 2007) and can produce reliable results with even low number of occurrences (Papeş and Gaubert 2007, Peterson et al. 2007, Wisz et al. 2008). One assumption of MaxEnt, and distribution models in general, is that the entire area of interest has been uniformly sampled (Phillips et al. 2009, Royle et al. 2012, Merow et al. 2013). Like other modeling methods, MaxEnt is sensitive to sampling bias, mainly due to unbalanced sampling effort across the area of interest (Phillips et al. 2009, Elith et al. 2011, Kramer-Schadt et al. 2013). Model quality can be severely affected if entire parts of the suitable area for a species are absent or if some areas are over-represented due to locally high sampling efforts (Fourcaude et al. 2014, Botella et al. 2020), often leading to spatial autocorrelation among observations (Phillips et al. 2009, Stolar and Nielsen 2015). As a result, the inaccuracy of the projections from biased models can seriously undermine the provision of reliable information for conservation purposes (Yackulic et al. 2013). Several ways to address this problem, which mainly differ depending on whether the sampling effort is known or unknown, have been discussed (Merow et al. 2013; Fourcaude et al. 2014). If presences are restricted to an over-sampled part of the study area, the model performance can be enhanced by sampling the background points from this part of the area (Phillips et al. 2009, Merow et al. 2013), i.e., the background points have the same sample selection bias as presence points. In this way, when the model projection is transferred to the rest of the area, the reliability of the predictions should be improved. In this case, the sampling design of *C. pullus* introduced a sampling bias: despite the homogeneous distribution of transects, the data were often highly concentrated in small and more accessible parts of the study area, mainly due to limitations on operator movement caused by running water. However, the georeferencing of the transects allows us to precisely know the sampling effort across the space. Therefore, in the MaxEnt implementation, the background sample was geographically restricted to all the 10 × 10 m cells crossed by the transects, assuming that these reflect most of the environmental conditions present in the whole study area. The cells in which at least one individual of the species was found were used as presences. The environmental variables used were DEM, TPI, TRI, and NDVI of 8 July 2022, as

this acquisition date was concurrent with the study period. The absence of high correlation between these variables ($|r| < 0.7$) was checked with the usdm package (Naimi 2017) in R. The modeling was performed with the package Biomod2 v. 4.2-2 (Thuiller et al. 2021) in R, randomly splitting the dataset into 70% for training and 30% for testing and performing 10 replicates of the procedure. The other parameters were kept by default. The modeling performance was evaluated using the Area Under the Curve (AUC) and the True Skill Statistic (TSS). Models with AUC below 0.6 are considered not reliable; 0.6–0.7, poor; 0.7–0.8, adequate; 0.8–0.9, good; and higher than 0.9, excellent (Araújo et al. 2005). Models with TSS measures below 0.4 are considered poor; 0.4–0.8, useful; and higher than 0.8, good–excellent (Allouche et al. 2006). An SDM was created averaging the replicates and then projected on the study area, producing a suitability map of the species (increasing suitability from 0 to 1000) and its transformation into a binary map of presence–absence (0 for absence and 1 for presence) according to the threshold that maximizes the TTS (Thuiller et al. 2021). This last map was used to estimate the amount of the actual predicted presence area in both the entire study area and in protected areas of the Natura 2000 network using QGIS.

Connectivity modeling.—Circuit theory was used to identify the different levels of connectivity and channels between patches along the study area (McRae et al. 2008), i.e., the most critical areas where connectivity bottlenecks and interruptions were present. Omniscape.jl software package (Landau et al. 2021) was used in Julia ver. 1.8.1 (Bezanson et al. 2017) to perform calculations, the inverse of the suitability map was used as the resistance surface, and the patches having suitability values above the threshold that maximizes the TSS metric were used as focal points. The maximum dispersal distance of the species was set as radius, i.e., 250 m (Rösti 2008, Maag et al. 2013).

Species persistence time.—The species persistence time is defined as the time span between the colonization and extinction of a species in a defined area (Bertuzzo et al. 2011, Suweis et al. 2012). In this study, this concept was applied to estimate the time between the colonization and abandonment of a patch by *C. pullus*, i.e., the time until the suitable NDVI range for the species is reached and then surpassed along a succession. For this purpose, an NDVI time series was used to calculate the vegetation change in each presence cell. Therefore, in addition to the 2022 NDVI map obtained to run the SDM, other NDVI layers were obtained starting from the data recorded from 2016 to 2021 by Copernicus project SENTINEL 2A and 2B (available at https://finder.creodias.eu), selecting layers as close as possible to the 2022 acquisition day (July 8). When needed (product type L1C), atmospheric corrections were made with SCP in QGIS. Only high-quality acquisitions with no cloud cover over the presence cells were retained. Acquisitions corresponding to July 2021 did not meet this requirement. Consequently, the final time series of NDVI layers includes the satellite data acquired on 24 July 2016, 4 July 2017, 9 July 2018, 4 July 2019, 8 July 2020, and 8 July 2022.

We assumed that during an undisturbed succession, the vegetation greenness on a floodplain may have a linear relationship with time due to the progression toward plant colonization and more productive plant communities. A simple linear model was therefore constructed for each presence cell using NDVI values as the dependent variable and the year of their acquisition as the independent variable. Only the reliable regressions were retained, i.e., those with p-value < 0.05 and $R^2 > 0.6$. These regressions were interpreted as linear relationships between vegetation greenness and time in the corresponding cells, on which no or negligible further disturbances occurred over the considered time span (7 years). As reported by Caponi et al. (2019), after a flood event, a vegetation succession can successfully evolve only if no further substantial disturbances occur, and vegetation recruitment is dependent on the morphology of the bars, seed dispersal, and the morpho-dynamics of the riverbed. The slopes of the retained regression lines were then used to estimate how many years from bare soil (NDVI = 0) are necessary to reach and surpass the suitable NDVI range for the species. This operation was performed by dividing the mean, the mean plus and minus one, and two standard deviations of NDVI in the overall presence cells by the mean slope and the mean slope plus and minus one standard deviation.

Results

Species distribution and connectivity.—Transects were covered from July 2 to July 10, intercepting 1398 10 × 10 m cells (Suppl. material 1: figs S1, S2). The presence of at least one individual of *C. pullus* was recorded in over 266 cells. The species was detected in all Natura 2000 sites intercepted by the study area—IT1110042, IT1110026, and IT1110053—and the observed individuals were mostly adults, except for two nymphs. The observations were made at elevations ranging from 1000 to 1910 m on sparsely vegetated gravel banks (Fig. 1) with a mean NDVI of 0.28 (SD = 0.06). The presence cells were mostly covered by bare soil with sparse vegetation composed of early successional plant species. The most frequent grass was *Calamagrostis pseudophragmites* (Haller f.) Koeler, especially below 1500 m of elevation, which constituted sparse formations or were in isolated tufts. On the occupied banks in more advanced stages of succession, there were also bushes species, particularly *Salix* sp. L., *Myricaria germanica* (L.) and *Hippophaë rhamnoides* L. and Desv., at various stages of development. Finally, on the terraces furthest from the recent flood events, *C. pullus* was occasionally found in patches with sparse small trees of *Pinus sylvestris* L., but even in this case, the portion of bare soil was abundant.

According to the metrics used to evaluate the SDM efficiency, the mean AUC and TSS values of the replicates were 0.86 (SD = 0.02) and 0.63 (SD = 0.01), respectively. Therefore, the SDM predictions were considered sufficiently reliable for subsequent analysis (Allouche et al. 2006, Araújo et al. 2005). The NDVI was largely the most informative environmental variable (Suppl. material 1: figs S3, S4). The predicted presence area obtained by the transformation of the suitability map to a binary map was 0.58 km^2, fragmented into 881 patches. The patch size was highly variable (mean = 655.73 m^2, SD = 1915.31 m^2).

The connectivity maps showed many geographic bottlenecks along the landscape. Furthermore, many interruptions along the river course were present, i.e., sectors along the study area where there was a distance greater than 250 m among nearest neighbor suitable patches. The current conservation area of the species was visualized by summing the predicted presence patches with the pixels of high connectivity importance, derived from binarizing connectivity maps using the value corresponding to the top third of the range as the threshold (Fig. 2, Suppl. material 1: figs S5, S6).

Species persistence time.—A total of 266 linear regressions were performed using the NDVI time-series values extracted for each presence cell of *C. pullus*. From these, 115 regressions (43.2%), defined

as those having a p-value < 0.05 and an $R^2 > 0.6$, were retained (Fig. 3). These regressions were interpreted as corresponding to cells where plant succession had not encountered substantial disturbances during the timespan considered, while in the others, there was no linearity, perhaps because the succession was disturbed by floods or human activity. Their mean slope of 0.0346 (SD = 0.0128) was used to estimate the persistence time of the species from flood disturbance by dividing the mean, the mean plus and minus one, and two standard deviations of NDVI by this value (Fig. 4, Table 1).

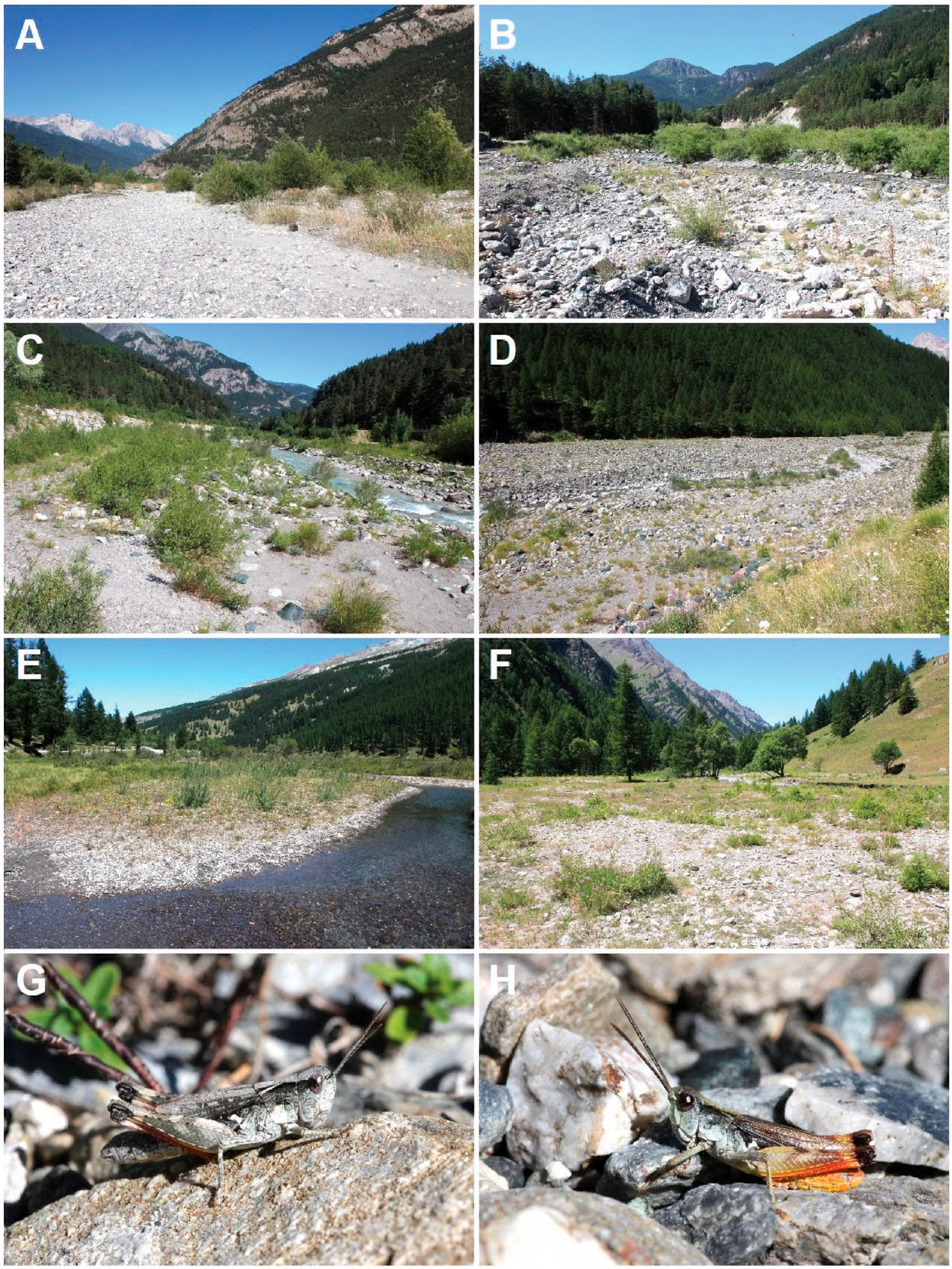
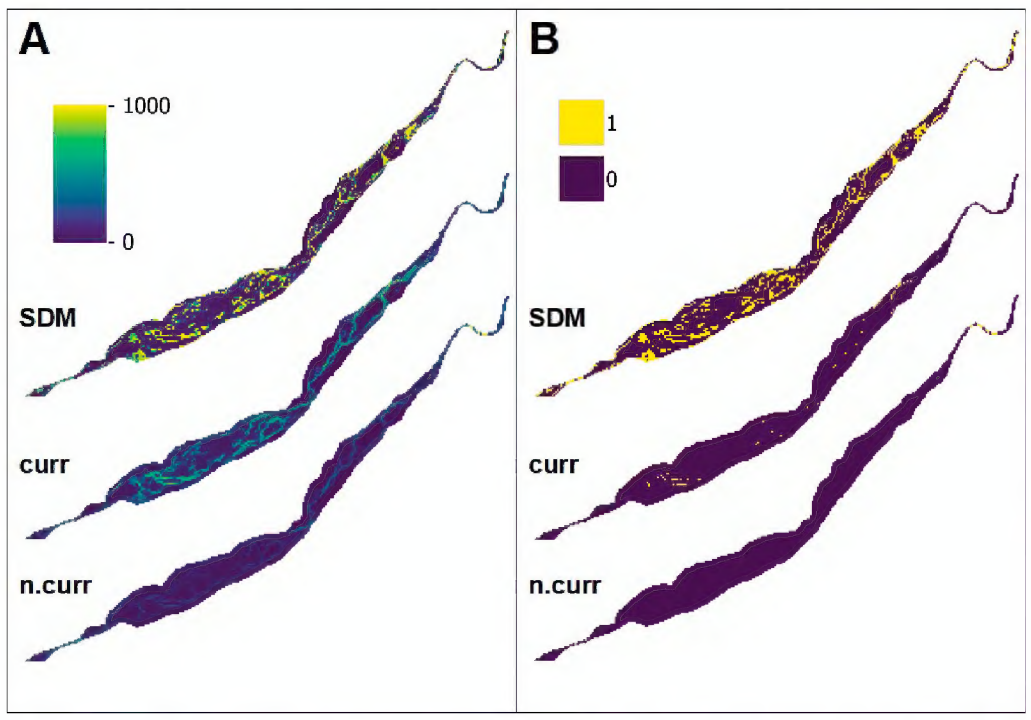


Fig. 1. Presence biotopes along Dora Riparia River. **A.** Oulx-Salbertrand floodplain at 1050 m; B. Cesana Torinese at 1260 m; **C.** Cesana Torinese inside IT1110026 at 1380 m; **D.** Sauze di Cesana floodplain at 1560 m; **E.** Sauze di Cesana near IT1110053 at 1840 m; **F.** Sauze di Cesana at 1900 m. A female (**G**) and a male (**H**) adult of *C. pullus* in Sauze di Cesana floodplain.

Discussion

According to the sampling results, *C. pullus* was more common in the upper Susa Valley than expected, as its presence in this area was only reported in two localities in 2007 and 2011 (Sindaco et al. 2012). To date, there are no observations available in the GBIF and iNaturalist databases websites (last check on 31 December 2023).



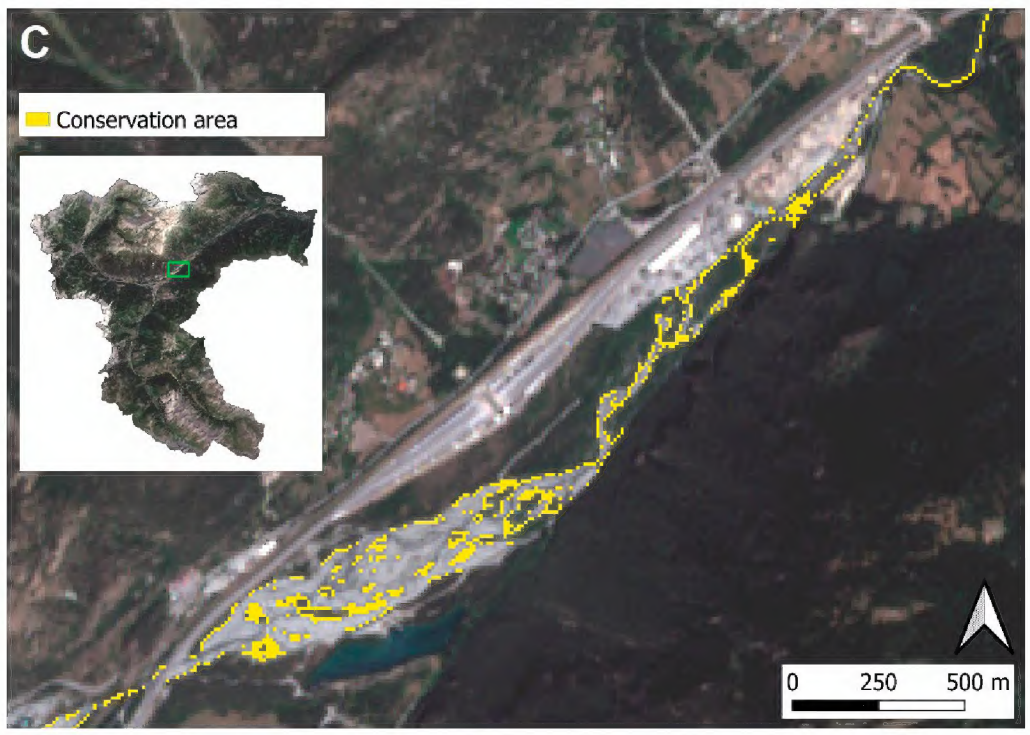


Fig. 2. Example of the identification of important conservation areas of *C. pullus* along the Dora Riparia floodplain in the municipalities of Oulx and Salbertrand. **A.** Species distribution model (SDM), cumulative current (curr), and normalized current maps (n.curr); **B.** Their binary transformation; **C.** A conservation area, based on the sum of binary transformations.

Table 1. Colonization, persistence, and abandonment along vegetational succession by *C. pullus*. For each NDVI value, the time to reach it is given. Values of years in square brackets were derived by dividing the NDVI value by one standard deviation above and below the mean slope of regression lines.

Value	NDVI	Years
min	0.02	0.57 [0.14, 0.90]
mean - 2SD	0.16	4.57 [3.34, 7.26]
mean - SD	0.22	6.28 [4.58, 9.96]
mean	0.28	7.98 [5.83, 12.67]
mean + SD	0.34	9.69 [7.07, 15.38]
mean + 2SD	0.39	11.40 [8.32, 18.09]
max	0.41	11.90 [8.68, 18.88]

The upper Susa Valley constitutes one of the three presence areas of *C. pullus* in the western Alps (Sindaco et al. 2012, Tami et al. 2021). Unfortunately, in one of these (Conca del Prà, Val Pellice TO, Baccetti 1958), the species ceased to be observed after 1950 (Tami et al. 2021). For this reason, the upper Susa Valley plays a fundamental role in the conservation of *C. pullus* in NW Italy.

In the study area, *C. pullus* was observed at up to 1910 m in elevation (i.e., at higher elevations than in previous records reported for the western Alps) but close to the maximum reported by Tami et al. (2021) for the eastern Alps (i.e., 2000 m). Its distribution was limited to sparsely vegetated habitats (Fig. 1), as also evidenced by the low greenness in the presence cells (mean NDVI of about 0.3). The dominant grass in the presence patches was *C. pseudophragmites*, which is typical of habitat 3220 protected by Habitat Directive 92/43/EEC (HD). *M. germanica* and *H. rhamnoides* were also observed in several presence patches; these species are characteristic of the protected riverine habitats 3230 and 3240, respectively. Further research is needed to identify and map the possible presence of these protected habitats in the floodplains of the upper Susa Valley.

According to the evaluation metrics, the SDM was highly reliable, and the importance of the NDVI in the modeling was disproportionate when compared to the other variables. This vegetation index, derived from remote sensing, can therefore be considered useful in this case, despite the resolution of the variables used, which, although high, can lose information on very small patches. In fact, Fontana et al. (2004) stated that this grasshopper can sometimes occupy patches of just 20 m^2.

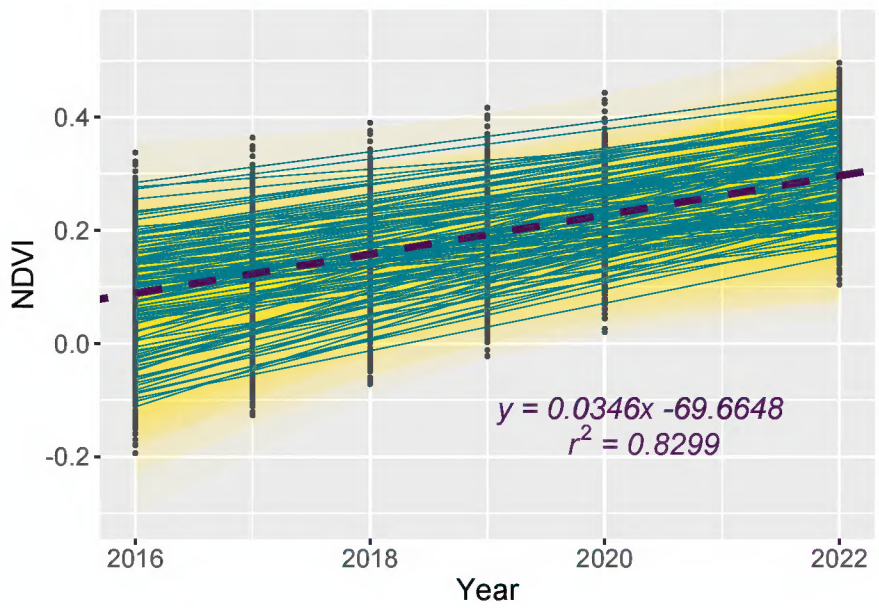


Fig. 3. Overlay of the regression lines (green) and their confidence interval (semi-transparent yellow shading). The dashed line represents the mean of the regression lines.

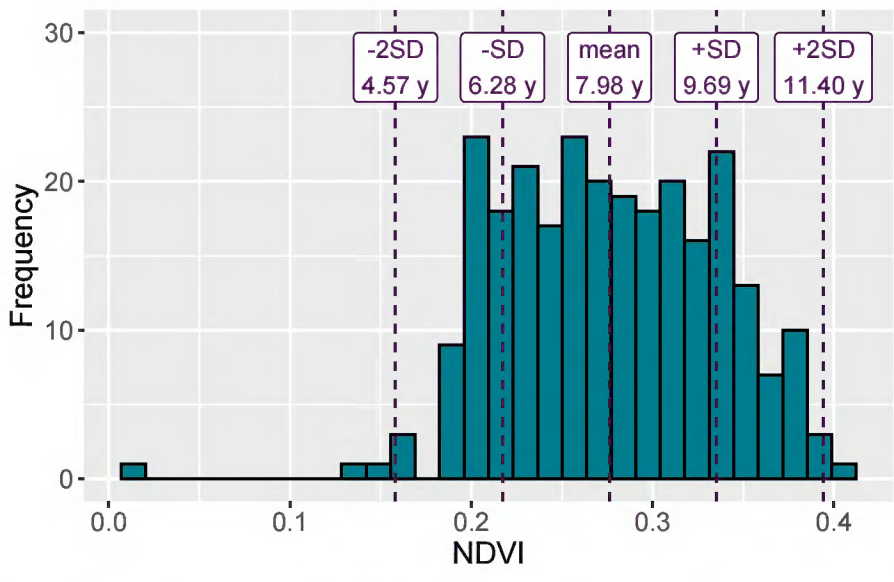


Fig. 4. Histogram of NDVI values in the overall presence cells. Labels indicate the estimated years necessary to reach the corresponding value.

The actual predicted presence area covers about 23% of the study area, with patches of varying sizes. Most of these are concentrated in large floodplains, where the river can flood more freely (Fig. 2). Geographic bottlenecks were uniformly distributed but more common in urban areas where rivers have been regimented between narrow, artificial banks. The species was found within three sites of the Natura 2000 network—IT1110042, IT1110026, and IT1110053—but only about 15% of the conservation area (i.e., the sum of the predicted presence area and the important connectivity areas) is included within these protected areas. The borders of the other three Natura 2000 sites—IT1110052, IT1110040, and IT1110010—were a few tens of meters away from suitable patches for the species:.

The time series of NDVI indicated a constant increment in vegetation greenness in almost half of the presence cells, as a positive mean slope of the reliable regression lines was found (Fig. 3). However, a significant relationship between NDVI and time was not detected in most presence cells. It can be hypothesized that in more than half of the presence cells, there may have been flood disturbances that interrupted the succession over the 7 years considered. The estimate of the persistence time of *C. pullus* showed the possibility of colonization of the banks within one year from bare soil. In the absence of further disturbances during succession, this species should be able to persist up to at least 12 years, and the most suitable conditions come about between approximately 5 and 10 years after flood (Fig. 4). It is interesting that also, according to Caponi et al. (2019), an increase in occupied surface by pioneer vegetation was recorded up to about 9–10 years from the beginning of an undisturbed succession on the Alpine Rhine River (Switzerland). Beyond this time limit, the pioneer vegetation decreased, leaving space for denser vegetation. As also reported by Corenblit et al. (2020) for the Isère River (FR), the massive sediment trapping carried out by the pioneer vegetation leads to the maturation of willow–poplar forest in approximately 10 years. The beginning of the early forest phase was similarly estimated by Politti et al. (2014) on Drau River (AT) banks (i.e., after at least 10 years from bare soil).

When this information is used to interpret the results of the current study, the overcoming of the temporal threshold of about 12 years from bare soil may indicate the achievement of an unsuitable succession stage with excessively dense vegetation for the persistence of *C. pullus* in a patch. Therefore, the survival of this species in the study area may depend on both the intermittent flooding, capable of disturbing the succession in the patches before reaching this threshold, and the existence of wide floodable areas, hosting a complex mosaic of patches with different discharge return frequencies. Natural floodplains play a key role in the long-term conservation of *C. pullus*, guaranteeing the simultaneous presence of source areas and areas to be colonized by this species. According to the study conducted on the River Sense (CH) by Gostner et al. (2010), gravel bars hosting *C. pullus* showed a specific inundation frequency of about 4–5 years, while bars without vegetation typically experience flooding with a frequency of less than two years. More densely vegetated banks instead occur with floods exceeding the five-year return period, which also coincides with floods primarily responsible for altering the riverine landscape. According to this information, sectors of rivers under flow regulation (e.g., with artificial riversides) offering narrow riverbanks subjected to high-frequency flooding may therefore give little chance of colonization to pioneer species. As also emerged from the models obtained in this study, the bottlenecks and the longest unsuitable stretches are mainly inside or near urbanized areas, which significantly reduces the connectivity between the populations present in the large floodplains.

According to future projections, flooding frequencies could undergo alterations caused by climate change in the Alps. The return period of extreme flood events (>100 years recurrence interval) could decrease significantly in many basins (Allamano et al. 2009, Wilhelm et al. 2022), while the frequency of large flood events (≥10 years recurrence interval) could significantly decrease (Wilhelm et al. 2022). The future survival of *C. pullus* and other riverine species could be at risk due to these changes, which are capable of compromising the balance between patches in different successional stages. A species can, in fact, survive in a patch only if the overall rate of extinction is compensated for by an equal overall rate of colonization (Levins 1969, Hanski 1991). In future scenarios, to survive extreme floods, *C. pullus* would therefore need source areas sufficiently away from more floodable areas and a sufficient time interval between floods to recolonize the lost areas. The decrease in the frequency of small floods could instead favor the expansion of dense vegetation in now flooded channels (Politti et al. 2013), leading to unsuitable conditions for these species. These future perspectives confirm that the maintenance of wide floodable areas should continue to be the main conservation strategy for this species, guaranteeing the constant availability of suitable patches over time through adequate temporal and spatial fluctuations in water level (Maag et al. 2013).

Conclusions

This study highlights the presence of an important population of *C. pullus* in the upper Susa Valley, mainly distributed in the floodplains along several kilometers of the Ripa and Dora di Bardonecchia rivers. As known throughout its range here, *C. pullus* represents a pioneer species capable of colonizing habitats with gravel and sand accumulation hosting scarce, mainly herbaceous, vegetation. The persistence time of this species along the succession seemed rather short (Fig. 5), and the presence of geographic bottlenecks and poor connectivity areas was evident, especially where the rivers cross urbanized areas. The coverage of suitable habitats by protected areas was extremely low. As in the rest of its geographical range, the main threats to this species in the upper Susa Valley are hydrological system modifications that reduce connectivity, generate many bottlenecks, and allow for a low number of suitable riverbanks. Furthermore, gravel pits and frequent activities in some stretches of the riverbed disturb the natural colonization of pioneer vegetation, which clearly constitutes the main succession stage suitable for the species. The maintenance of large floodable areas and, in particular, the conservation of floodplains are key in its long-term conservation.

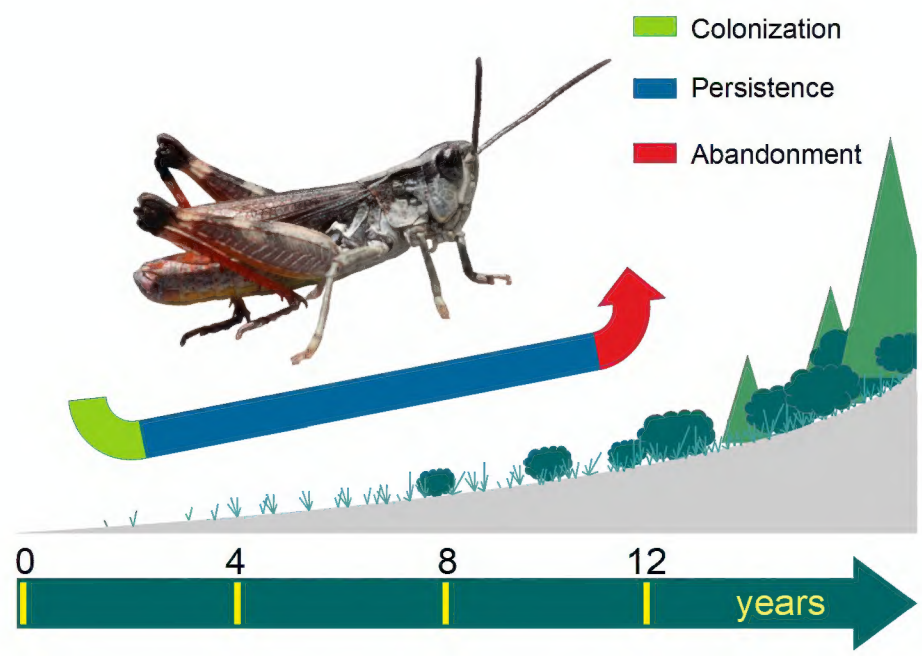


Fig. 5. Graphical representation of colonization, persistence, and abandonment of a patch along vegetational succession by *C. pullus*.

Although *C. pullus* is not protected by Italian or European legislation, planning for its conservation should be seriously considered to avoid its extinction in Northwestern Italy. Furthermore, as an indicator of riverine environments with high natural value (Reich 1991, 1998, Fontana et al. 2002, 2004, Massa et al. 2012, Tami et al. 2005, 2021), its presence contributes to the identification of important areas dominated by fluvial dynamics capable of maintaining a high diversity of habitats and species that have adapted to the natural flood pulsations. Indeed, this species should be considered an umbrella species (Frankel and Soulé 1981), as its protection produces positive effects on other species living in the same habitats (Fontana et al. 2004). In accordance with the purpose of this study, the conclusions include the following recommendations and policy suggestions for the long-term conservation of *C. pullus* in the upper Susa Valley along the Dora Riparia and Dora di Bardonecchia rivers (Fig. 6):

C. pullus should be included among the "other important species of flora and fauna" in the standard data forms of the IT1110042, IT1110026, and IT1110053 Natura 2000 sites to allow for consideration of this species in impact assessments relating to projects involving riverine habitats.

The Natura 2000 IT1110052, IT1110040, and IT1110010 sites should be extended downwards to include the nearby floodplains and constrained reaches in order to facilitate the conservation of suitable habitats

Efforts should be made to maintain the natural river dynamics and, when possible, restore them, expanding the planned floodable surfaces to promote the creation and maintenance of a diverse mosaic of riverine habitats.

Patches with natural accumulations of gravel and sand of at least 20 m^2 along regulated river sections, spaced at a maximum distance of 250 m from each other, should be reserved when pos-

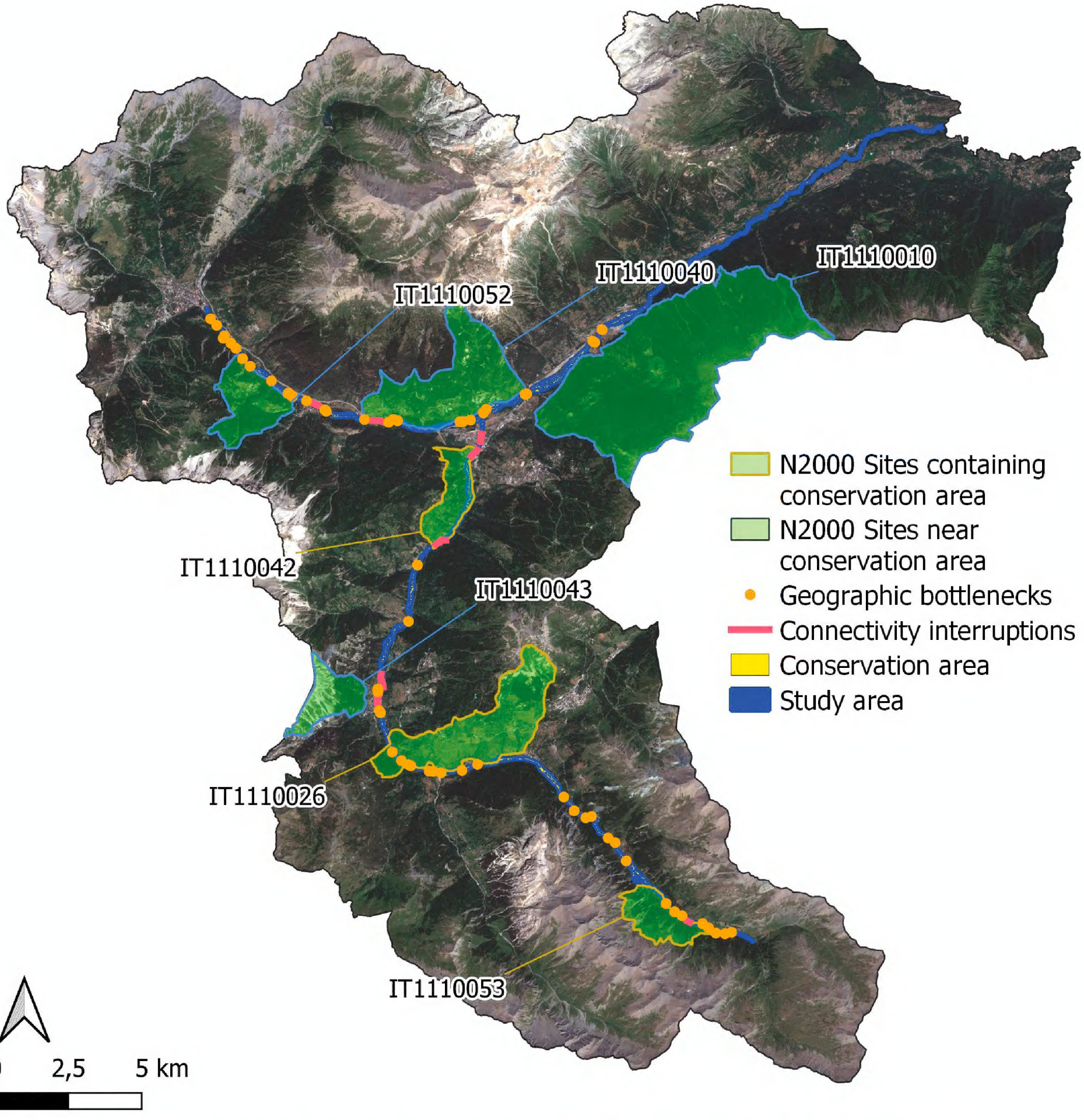


Fig. 6. Summary of the geographical information useful to plan the conservation of *C. pullus* in upper Susa Valley.

sible to create small suitable habitats that improve the species' connectivity between floodplains.

When gravel moving or extraction on floodable surfaces are inevitable, the following conditions should all be met:

The space for works should be limited: The extension of a single disturbed surface should not exceed 100 m^2, and it should be interspersed with intact and suitable sparse vegetated areas that act as a source of recolonization for the species.

Works should be limited in terms of time: Works should not last more than a year, thus simulating a single flood event capable of creating bare soil that can be colonized by pioneer species.

Works should be limited in terms of succession: Works should be carried out only on patches that present a stage of succession with unsuitable vegetation for the species, i.e., on patches where at least 12 years have passed since the last disturbance (natural or artificial) that created bare soil. Works performed in accordance with this condition could also promote the survival of the species in a future scenario of flood reduction.

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Supplementary material 1

Author: Luca Anselmo
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